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Non-Native Species

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KEY HEADLINES

• Climate change has facilitated the establishment and spread of certain marine non-native species (NNS) in the North-east Atlantic (e.g., Pacific oyster *Crassostrea gigas*).

• Evidence now exists to demonstrate that indirect consequences of climate change, such as ocean acidification and increased extreme weather events will favour certain NNS.

• The distribution and dispersal rates of marine NNS in the UK and Ireland are now far better understood, which will enable greater confidence in predicting the future effects of climate change on these species.

• The training of future taxonomists is vital, and the development of new technology, such as environmental DNA (eDNA) may offer new opportunities for the accurate plotting of species' ranges through time.

• Providing definitive evidence of any direct linkage between climate change and the spread of the majority of NNS is extremely challenging, due to other confounding factors, such as anthropogenic activity, natural secondary spread and the lack of research studies on cause and effect.

• The introduction of NNS via marine debris and the opening of links between the Pacific and Atlantic Oceans due to melting of Arctic sea ice, the increasing risk of non-native disease introduction and the rise in religious releasing of NNS into coastal environments, were all identified as significant emerging issues. In addition, increasing seawater temperatures are likely to facilitate the further spread of many NNS throughout the UK.

• Predicting which NNS are likely to arrive and cause a significant environmental and/ or economic impact, and how this is likely to be affected by the direct and indirect consequences of climate change, is still extremely challenging and will require urgent research effort over the next decade.

1. INTRODUCTION

Climate change and non-native species (NNS), i.e. those that have been intentionally or unintentionally introduced, as a consequence of human activity, from outside their native range (CBD, 1992), individually present two of the greatest global threats to biodiversity and the provision of ecosystem services (Halpern *et al.*, 2008; Burgiel and Muir, 2010; Brodie *et al.*, 2014). Once introduced, certain NNS can become established (i.e., grow and reproduce successfully) and their subsequent dispersal from the point of introduction can result in significant economic impacts on terrestrial, freshwater and marine environments (i.e., 'invasive' NNS – species that have a significant negative impact on the environment or the economy). In the United States alone are estimated at $120 billion annually (Pimentel *et al.*, 2005), while in the UK the annual estimate is £1.7 billion (Williams *et al.*, 2010). Considerable growth in international trade (Hulme, 2009) and intentional stock movements for mariculture purposes (Mineur *et al.*, 2014) have caused an exponential increase in the spread of marine NNS around the world and the progressive banning of tributyltin in antifouling paints globally over the last decade can only exacerbate the problem (Mineur *et al.*, 2007). This trend of increasing rates of arrivals and greater secondary spread (i.e., spread that occurs either by natural dispersal or by anthropogenic vectors following initial introduction) has been observed in the UK throughout the 20th Century (Minchin *et al.*, 2013; Bishop *et al.*, 2015b) (Figure 1), although this figure may have been slightly inflated over the last 10 years due to increased research activity on the subject.

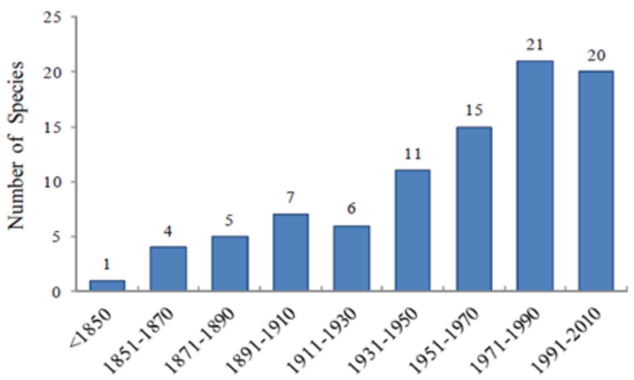

Figure 1. *Number of non-native species introduced into brackish and marine environments in British waters over 20 year intervals from <1850 to 2010 (N=90) (Minchin et al., 2013).*

Global sea surface temperatures have increased in the last few decades (IPCC, 2014). Regional climate models predict that there will be a continuation in the current trend of warming throughout the 21st century (IPCC, 2014). From 1974 to 2010, the general trend has been a 1-2 °C increase in sea surface temperatures (SST), with the greatest rises in temperature experienced on the East coast of the United Kingdom compared with lower increases in SST on the Western coasts of Ireland and Scotland (Yesson *et al.*, 2015). Furthermore, an increase in the magnitude and frequency of short-lived high temperature events is predicted (Meehl *et al.*, 2007), which could lead to irreversible changes, particularly in marine communities (Sorte *et al.*, 2010; Wernberg *et al.*, 2016). There is rapidly growing evidence, particularly in the Mediterranean, that rising seawater temperatures are both increasing the potential for NNS invasion, establishment and subsequent spread, and exacerbating the impacts of these species on native species and communities (Galil *et al.*, 2015).

In the North Atlantic, human-mediated changes to climate and extreme weather events have already led to the spread of NNS, following their initial introduction by anthropogenic vectors (Reid *et al.*, 2009; Pederson *et al.*, 2011). Climate change can influence every stage of the invasion process following initial introduction, including establishment and secondary spread (Occhipinti-Ambrogi, 2007; Maggs *et al.*, 2010; Pederson *et al.*, 2011). It is often difficult, however, to directly link climate change to particular observations such as, shifts in NNS distribution or increases in reproductive success, owing to a lack of biological and physiological data (Pederson *et al.*, 2011). There is strong evidence, however, based on a combination of observational studies and a greater understanding of temperature tolerances of some species to indicate direct effects of increasing seawater temperature. For example, the influx (i.e., recruitment) of new juveniles of the Pacific oyster *Crassostrea gigas* and their subsequent survival to mature individuals at more northerly locations in the UK has been attributed to warming seawater (Syvret *et al.*, 2008; Guy and Roberts, 2010; Kochmann *et al.*, 2012; Cook *et al.*, 2014). Increased seawater temperatures, may also have contributed to the establishment of other NNS, such as the ruby bryozoan *Bugula neritina*, which has now established populations outside the warm water areas created by power station outflows where it was initially observed (Ryland *et al.*, 2011).

Ninety NNS have been identified in British and Irish marine and brackish environments (Roy *et al.,* 2012; Minchin *et al.*, 2013; Minchin and Nunn, 2013), although, at least another 10 species have been reported since these reviews, including the compass seasquirt *Asterocarpa humilis* (Bishop *et al.*, 2013), the orange ripple bryozoan *Schizoporella japonica* (Ryland *et al.*, 2014) and two species of Asian shore crab (*Hemigrapsus sanguineus* and *H. takanoi*) (Seeley *et al.*, 2015; Wood *et al.*, 2015a). The majority of marine NNS in Britain originate from the North Pacific, followed by the Northwest Atlantic, where environmental conditions are typically compatible with the UK (Minchin *et al.*, 2013; Mineur *et al.*, 2015a). Many are initially reported at sites of anthropogenic activity, such as ports, marinas and aquaculture facilities, particularly in the vicinity of the English Channel, with a number subsequently spreading northwards (Minchin *et al.*, 2013; Bishop *et al.*, 2015b).

Understanding the role that changes in seawater temperature and other indirect consequences of climate change play in the initial establishment of NNS and the range that they may occupy is critical, if the environmental and economic impacts of NNS are to be mitigated (Bohn *et al.*, 2012). The aim of this decadal review is to provide an update on how climate change is influencing how NNS become established, disperse and then potentially impact on native species and habitats in the coastal and marine environment around the UK and Ireland. We also consider how our understanding on this subject has developed over the last 10 years, to highlight the knowledge gaps/ current challenges and to discuss emerging issues in this field with respect to the present time and in the future.

2. TOPIC UPDATE

A consequence of anthropogenically-induced climate change is that the physiological limits of some species will be challenged, whilst others will thrive. Climate change may influence marine NNS invasions in four ways:

1. Established warm-water NNS may expand their invaded range northwards in the boreal hemisphere, as higher latitudes increase in temperature and either out-compete native cold-adapted species or fill the niches left behind as the latter species track their temperature niche north (Occhipinti-Ambrogi, 2007).

2. New warm-water NNS may become established as a consequence of environmental conditions, as the recipient environment becomes more favourable (Wrange *et al.,* 2010).

3. A lowering in ocean pH, as a result of increasing atmospheric CO_2, may indirectly facilitate the introduction and establishment of NNS through alterations in trophic dynamics (Hulme, 2005).

4. An increase in extreme events (i.e., increased storms, heat waves, precipitation, etc.) may favour NNS that already have traits which enable them to cope with environmental thus enabling them to out-compete native species (Rogers & McCarty, 2000; Lenz *et al.*, 2011).

2.1. Northwards expansion of currently established NNS in the UK and Ireland

The most widely documented of these four proposed changes over the last few years has been the general northwards expansion of a number of marine NNS. Many marine NNS have entered the UK at sites in southern England, so their subsequent range expansion is bound to have a strong northward component. However, there is indirect evidence suggesting that this range expansion is linked to increased seawater temperatures, although further experimental and field studies are required to confirm a definitive link. In recent years, species that have been documented as expanding their range in the UK, potentially in response to warming SST include the Pacific oyster *Crassostrea gigas* (Cook *et al.*, 2013a), the alga *Pikea californica* and the clubbed tunicate, *Styela clava* (Nunn & Minchin, 2009).

CASE STUDY – The Pacific oyster Crassostrea gigas

Increasing seawater temperatures are highly likely to have contributed to the recruitment of the commercial Pacific oyster *Crassostrea gigas* (Syvret *et al.*, 2008; Guy & Roberts, 2010) in natural habitats at a number of more northerly locations in the UK (Kochmann *et al.*, 2012; Cook *et al.*, 2014) and Europe (Wrange *et al.*, 2010), as well as the extensive expansion of this species at existing sites (Herbert *et al.*, 2012). In 2014, a survey was undertaken in Scotland to determine the extent of wild *Crassostrea gigas* following sightings of this species in the Firth of Forth, SE Scotland (Smith *et al.*, 2014). From the 60 sites surveyed, live wild *C. gigas* were found in five locations on the north shore of the Solway Firth (SW Scotland) (Cook *et al.*, 2014) (Figure 2). The majority of *C. gigas* found in the survey were between 50 – 100 mm in length, comparable in size to those found in the Firth of Forth (Smith *et al.*, 2014). This survey determined the current northern most limit for this species on the west coast of the UK (54.8° N). As more northerly populations are recorded, however, in Ireland (55.1° N) (Kochmann *et al.*, 2013) and Scandinavia (60.0° N) (Wrange *et al.,* 2010), it is highly likely that *C. gigas* will continue to spread further northwards (Cook *et al.*, 2014).

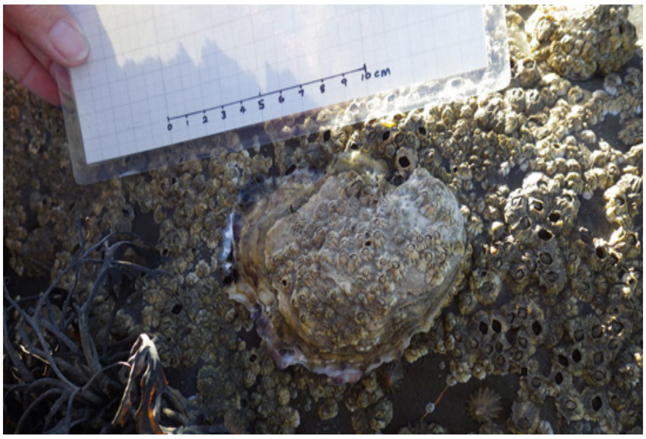

Figure 2: *The Pacific oyster Crassostrea gigas recorded from the intertidal zone, south-west Scotland in 2014. © Christine Beveridge, SAMS*

CASE STUDY – Red macroalga Pikea californica

Pikea californica originates from the west coast of North America and was likely introduced to the Isles of Scilly, an archipelago of islands located 45 km off south-west England, where it has been established since at least 1967. Searches in Cornwall, south-west England in the 1990s failed to detect this species and it was thought that the Isles of Scilly may have been particularly favourable because of their unusually warm winter seawater temperatures (ca. 9 °C min.) (Maggs and Guiry, 1987; Maggs and Ward, 1996). In January 2015, *P. californica* was found in Newlyn Harbour, on the south-west coast of Cornwall (D. Fenwick, pers. comm.), where the difference between winter and summer mean monthly temperatures from the Isles of Scilly is only 5 °C. At present, however, the mean SST for February has increased to 10 °C in south-west Cornwall, which may have enabled the establishment of *P. californica* in this region, when previously, the colder winter temperatures would have prevented its' survival. A similar narrow temperature range to the Isles of Scilly, caused by enhanced mixing, also occurs off the NW coast of Cornwall (G. Savidge, pers. comm.), and this would be the most likely area in which to expect additional populations of *P. californica* to occur if the species is indeed spreading.

2.2 New establishment of 'warm-water' NNS as a consequence of changing environmental conditions

Certain NNS in GB appear to have remained restricted to widely-spaced artificially warmed sites, or to enclosed sites experiencing enhanced summer temperatures. Before a NNS spreads from the location in which it has become established, a delay is often observed. This may represent a demographic lag phase (Kowarik, 1995), perhaps accompanied by acclimatisation to local temperatures, but long-term restriction to sites with elevated temperatures strongly suggests that a requirement for warm water has prevented further spread. At least two of the NNS from the warmer regions of the world that were largely limited to a few, scattered, heated and/or enclosed sites throughout much of the 20th century were declared to have disappeared from GB with the decommissioning of power stations or removal of lock gates in their favoured locations, but have subsequently been re-discovered occupying a much broader geographical range. They are the bryozoan *Bugula neritina* and the barnacle *Amphibalanus amphitrite*. The warming trend in the sea has probably contributed to the re-emergence of these species following re-introduction, allowing survival and reproduction at open-sea temperatures and thus a much broader geographical range than previously achieved.

CASE STUDY – The ruby bryozoan Bugula neritina

Bugula neritina (Figure 3), is typically found in warm-temperate and sub-tropical coastal waters, and was largely restricted to a few, scattered, heated and/or enclosed sites throughout much of the 20th century in the UK. With the demise of these sites, the species was stated by Eno *et al.* (1997) to be 'No longer established in the wild'. *Bugula neritina* was subsequently rediscovered, however, occupying a much broader geographical range (Ryland *et al.*, 2011). Arenas *et al.* (2006) found *B. neritina* to be widespread on the south coast of England in 2004 and it was soon recorded widely on the English and Welsh coasts, in southern Ireland and as far north as the Firth of Clyde on the west coast of Scotland (Ryland *et al.*, 2011). Subsequent surveys have added records of *B. neritina* in Belfast and Carlingford Loughs, Ireland where it is locally abundant (Minchin and Nunn, 2013) and documented additional sites in Wales (Wood *et al.*, 2015b). The species, however, has remained unrecorded in recent surveys in NW England (Wood *et al.*, 2016) and has not been recorded north of Lowestoft on the east coast of England, where it was already present in 2009 (Bishop *et al.*, 2015b), or been reported from eastern Scotland. It was not found in northern Scotland by Nall *et al.*, (2014) or Collin *et al.*, (2015). *Bugula neritina* may thus now be close to its current northern limit in the UK after an apparent rapid expansion northwards.

Figure 3: *The ruby bryozoan, Bugula neritina © J.D.D. Bishop, MBA.*

CASE STUDY – The barnacle *Amphibalanus amphitrite*

The barnacle *Amphibalanus amphitrite* is a cosmopolitan species of subtropical and tropical waters. In the British Isles, the species existed during the 20th century in a few artificially warmed sites, but Southward (2008) could name no extant populations and noted that the species had not been seen since 1974 in one of its former locations, Plymouth. On the other side of the English Channel and southern North Sea, *A. amphitrite* similarly occurred at enclosed sites with artificially elevated temperatures, and in surveys conducted in 1997-1999, it was found frequently on navigation buoys off the Belgian coast (Kerckhof and Cattrijsse, 2001); the presence of gravid individuals on the buoys indicated the ability to reproduce at contemporary open-sea temperatures. In 2010-11, *A. amphitrite* was found on boat hulls in two Plymouth marinas and on a settlement panel in a third marina (J Bishop, C Wood and A Yunnie, pers. obs.), two of these sites being relatively open and not expected to experience locally elevated temperatures. Surveys of English harbours and marinas in 2014 documented *A. amphitrite* at five additional sites on the south coast (Wood *et al.*, 2015c), although four of these were lock-gated basins, possibly with enhanced summer warming because of limited exchange with the open sea. As with *Bugula neritina*, this species seems to have become re-established across a much broader range of sites than formerly known.

Changes in the prevailing climate, notably a rise in seawater temperatures, have also potentially assisted other non-native species and allowed their establishment in UK waters.

CASE STUDY – The kuruma prawn *Marsupenaeus japonicus*

Marsupenaeus japonicus was first reported in the English Channel by Clark (1990a, b) and it was considered that these specimens were possible escapees from one or more French hatcheries with brood stocks of this species. Recently Quigley *et al.* (2013) recorded the occurrence of 14 kuruma prawn specimens from the Celtic Sea, English Channel, and North-West France. *Penaeoidea* do not retain their eggs, but release them directly into the water. The larvae then undergo an extensive metamorphosis before becoming juveniles. Although adults can tolerate water temperatures down to 10°C (Lucas and Southgate, 2003), larval development requires temperatures above 24 °C. Any further rise in sea temperatures in the region of the western approaches to the English Channel could, therefore lead to escapees spawning eggs in the wild and potentially completing their complex life cycle.

CASE STUDY - Non-native plankton: results from the Continuous Plankton Recorder (CPR) Survey

The CPR Survey has been running with consistent methodology since 1958, identifying and enumerating both phytoplankton and zooplankton organisms from the North Atlantic, and more recently from the South Atlantic, Arctic and Pacific Oceans. This extensive temporal and spatial coverage provides an ideal baseline for rapid identification of new plankton arrivals in the areas covered.

Over the last 20 years, a number of non-native planktonic organisms have been identified around UK waters, both phytoplankton and zooplankton (Nehring, 1998). These organisms are hypothesized to have been anthropogenically introduced, via vectors such as ballast water (Edwards *et al.*, 2001). Below is a short summary of some plankton species that have recently been identified.

Pseudodiaptomus marinus, a small copepod (Crustacea) was originally described from eastern Asian waters, and later spread to the Indo-Pacific region. In 2011, the species was identified in CPR samples from the southern North Sea, and simultaneously in time-series net samples taken to monitor the German EEZ. The species is thought to have been introduced in ballast water from cable ships to Calais harbour, and then advected with prevailing currents along the Belgium coast. The species has since become established in the area (Jha *et al.*, 2013 and references therein).

Another planktonic crustacean, the cladoceran (water flea) *Penilia avirostris*, has become a common member of the late summer zooplankton community in the North Sea since the early 2000s. Prior to this time, the species was commonly found in the Bay of Biscay and further south, into the Mediterranean. The species is thought likely to have been introduced via ballast water (*Cladocera* commonly produce a resting egg stage via sexual reproduction), and increasing SSTs, particularly during autumn months, have led to its establishment. The species was originally identified in the North Sea by the CPR Survey, and also in samples taken at Helgoland Roads (Johns *et al.*, 2005).

2.3 Ocean Acidification

It has been estimated that the oceans, which are a major sink for anthropogenic CO_2, will undergo a decline in sea water pH by up to 0.5 units by 2100 (IPCC, 2007). Non-native species (and native) that are tolerant to low carbonate saturation in seawater are likely to benefit in seawater with an increased atmospheric partial pressure of carbon dioxide (p CO_2) (Brodie *et al.*, 2014). The introduced red seaweed *Neosiphonia harveyi* showed no decline in photosynthesis at low temperatures, as expected, under increased pCO_2, suggesting that this warmer water species might be able to spread northwards as ocean acidification intensifies in the future (Olischläger and Wiencke, 2013). In the Mediterranean, volcanic CO_2 vents, which lower the pH of the water column, were found to support a number of invasive macroalgae, including, *Sargassum*, *Caulerpa* and *Asparagopsis*, whereas native coralline algae were excluded (Hall-Spencer *et al.*, 2008). Evidence in the UK and Ireland, however, for the biological and ecological effects of ocean acidification on NNS under field trials are still lacking (Pederson *et al.*, 2011).

2.4 An increase in extreme events, as a result of climate change may favour NNS

An increase in extreme events, as a result of climate change can have both direct (i.e., provision of conditions which favour NNS over native species) or indirect effects in which measures to mitigate the impacts of these events (i.e., the development of additional flood defences) can provide suitable habitat for NNS to become established. One of the mechanisms by which NNS are understood to flourish in invaded habitats is their enhanced ability to tolerate stress compared to native species (Lenz *et al.*, 2011). Extreme climate events such as intense storms are likely to lead to stressful conditions, including pulses of low salinity water, physical disturbance, sediment resuspension and deposition, which could potentially give a competitive advantage to certain NNS, such as the colonial tunicate *Didemnum vexillum* (Gröner *et al.*, 2011).

CASE STUDY – The Asian shore crabs *Hemigrapsus takanoi* and *H. sanguineus*

An indirect consequence of climate change has been the development of hard, flood and erosion defences in the Wadden Sea, which have been partially blamed for the successful establishment of both *Hemigrapsus takanoi* and *H. sanguineus*. Both species are of concern due to the impacts they have had within their known invasive range, in particular through intertidal competition with the native shore crab *Carcinus maenas* (Landschoff *et al.*, 2013).

These two species of shore crab (Figures 4 and 5) were recorded in the UK for the first time in 2014. *Hemigrapsus takanoi* from the River Medway, Kent (and retrospectively in preserved samples from a *Crassostrea gigas* reef in the Colne, Essex in 2013) (Wood *et al.*, 2015a) and *H. sanguineus* from the shores of Glamorgan and Kent (Seeley *et al.*, 2015). Sea temperatures are currently well within the required parameters for *H. sanguineus* to become established, with mean summer sea temperatures >13 ˚C. This temperature was suggested by Stephenson *et al.* (2009) as the threshold below which the species was not found in Maine, USA. They also reported that mean population densities and reproductive activity increased as mean temperatures exceeded 15 ˚C and 12 ˚C, respectively. With current North Sea average summer SST ranging between 14 ˚C and 16 ˚C and a predicted future minimum temperature of 15 ˚C, it would appear likely that the species will benefit from predicted warming and become more abundant in this region. In addition, as storm damage and flooding (phenomena linked to climate change) increases in coastal areas, it is more and more likely that coastal defence structures will be used, with the potential to create additional refugia and suitable habitats for the Asian shore crabs.

Figure 4: *Asian shore crab Hemigrapsus takanoi © Jack Sewell, MBA.*

Figure 5*: Asian shore crab Hempigrapsus sanguineus © Jack Sewell, MBA.*

CASE STUDY – The kelp ‘Wakame’ *Undaria pinnatifida*

The kelp *Undaria pinnatifida* (Wakame) (Figure 6) is native to areas of the NW Pacific (the coastlines of Japan, Korea, Russia and China) and has been introduced to temperate areas worldwide. *Undaria pinnatifida* was first recorded in the UK in 1994 on floating marina pontoons in the Hamble Estuary (Fletcher and Manfredi, 1995) and by 1999, had expanded along the south coast of England (Farrell and Fletcher, 2006). Today, the species is present northwards on the east coast as far as the Firth of Forth, SE Scotland (J. Khan-Marnie, SEPA, pers. comm.) and on the west coast in North Wales, Lancashire and Glenarm, Northern Ireland (E. Cottier-Cook, pers. obs.).

Undaria pinnatifida can act as a pioneer species in its native environment, and is part of a natural successional colonisation process (Agatsuma *et al.*, 1997). Within its non-native range, this pioneer-like behaviour is highlighted by evidence that ecosystem stress or disturbance is key to the recruitment of *U. pinnatifida*. Native canopy disturbance or die back is often a critical factor increasing the spread and abundance of the non-native kelp in mixed canopy assemblages (Thompson and Schiel, 2012). Extreme climatic events are likely to exacerbate such disturbance and die back events, thus potentially allowing *U. pinnatifid*a to proliferate and spread within previously native-dominated kelp habitats (Brodie *et al.*, 2014).

In particular, reductions in salinity following prolonged rainfall may facilitate *U. pinnatifida* colonisation. This macroalga is generally found in fully saline conditions, with mean salinity values below 27 often considered to limit its range (Saito, 1975); however, it may survive periods of reduced salinity as low as 11 – 6 (Peteiro and Sánchez, 2012). Increased rainfall and storminess are predicted to cause periodic drops in salinity, particularly in enclosed environments such as marinas and harbours. Due to the ability of this species to survive during periods of low salinity, it is likely to persist in these affected areas.

How increased storminess will affect the distribution and abundance of *U. pinnatifida* remains unclear. Although this species is limited from highly exposed environments, which may become more common due to climate change, increased disturbance of native canopies from storms and low salinity events are likely to facilitate further spread and proliferation of this non-native kelp.

Figure 6: *Wakame – Undaria pinnatifida © J.D.D. Bishop, MBA*

2.5 CONFIDENCE ASSESSMENT

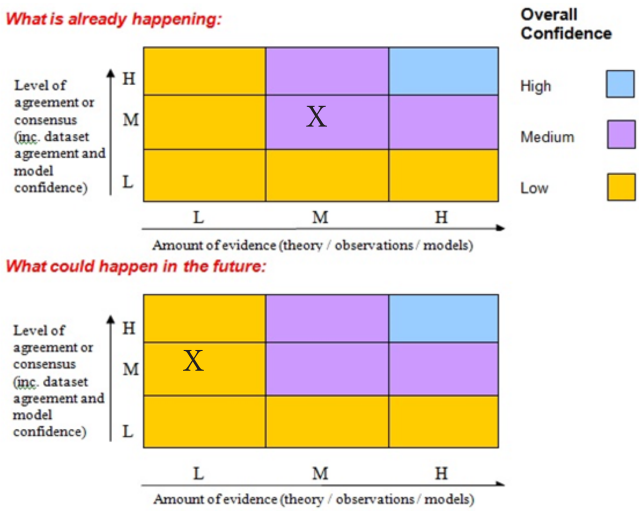

Figure 7: *Confidence assessment matrix*

The overall confidence levels shown above for both what is already happening and what could happen in the future have remained the same as the MCCIP report cards produced in 2010 (Maggs *et al.*, 2010) and 2013 (Cook *et al.*, 2013a) based predominantly on existing evidence and the level of agreement amongst experts in this field. In the 2013 report card, however, the level of agreement/ consensus did increase for what could happen in the future, from low to medium based on the reviews by Pederson *et al.* (2011) and Sorte *et al.* (2013) and there is consensus that this should remain as medium based on horizon-scanning work undertaken at both a GB (Roy *et al.*, 2014a) and European level (Roy *et al.*, 2015).

3. HOW OUR UNDERSTANDING HAS DEVELOPED OVER THE PAST DECADE?

Over the last decade, our understanding on NNS has made considerable strides, particularly in determining the baseline information on the rate of introductions, the presence/ absence and hence, the distribution of many species previously unknown in the UK and in predicting new arrivals to UK, as a consequence of either human mediated transportation and/ or climate change. Specific recording systems have also enabled us to interrogate the NNS datasets to identify trends and provide evidence-based policy and management advice.

3.1 Rate of NNS introductions

Following a major review of published and grey literature, the number of new NNS in the UK was found to have continued to increase since the 1930s, with approximately half having been recorded since 1970 (Figure 1). In the past 20 years, however, the rate of increase has plateaued with a similar number, i.e., ca. 20 new species recorded in the UK as in the previous 20-year period (Minchin *et al.*, 2013). Unfortunately, these data cannot be linked directly to any changes in the environment, as a consequence of climate change, but it does act as a significant contribution to the knowledge base for NNS in the UK.

3.2 Improvement in baseline NNS distribution data throughout the UK

The 2007–2008 Report Card on NNS noted under Knowledge gaps that: “One of the major problems of assessing the potential impact of climate change on non-native species is the lack of knowledge regarding where many of the species are established. There has been no full scale baseline survey of the presence of non-native species in the marine environment, so the current distribution of many species is not known.” A report by Stebbing *et al.* (2014) noted the continuing absence of a targeted national statutory programme specifically undertaking monitoring and surveillance of marine NNS in UK waters. In that report however, several existing non-statutory and statutory schemes were detailed that provide, or have the potential to provide, data on the occurrence of NNS.

A specific gap was noted by Stebbing *et al.* (2014): the absence of routine survey work targeting locations at high risk of introduction of NNS such as ports and marinas. Nevertheless, progress in this regard has been made over the last 10 years, when rapid assessment surveys (RAS) of harbours and marinas have been found to be an efficient and cost-effective way to gather synoptic, comparable data on the changing distributions and prevalence of many NNS around the coast, including the systematic recording of species’ absences (Arenas *et al.*, 2006; Ashton *et al.*, 2006; Minchin, 2007; Minchin and Nunn, 2013; Nall *et al.*, 2014; Bishop *et al.*, 2015b; Collin *et al.*, 2015; Wood *et al.*, 2015b; Wood *et al.*, 2016). The increasing adoption of this approach has led to a much improved documentation of the ranges of a substantial number of NNS. As is the case for rocky-shore monitoring programmes such as MarClim, the relatively detailed distribution information from RASs is critical for identifying future climate-driven changes and has been important in providing base-line data for several NNS in the UK.

On the English coast, the ambition has been to repeat RAS visits to the same sites every 2–4 years, which has provided insight into patterns of colonization and revealed contrasting rates of spread in different species (Arenas *et al.*, 2006; Bishop *et al.*, 2015a; 2015b; Wood *et al.*, 2015c; 2016). The overall RAS data gathered by different groups in the UK and Ireland jointly document unexpectedly rapid spread of some sessile NNS around our coasts, indicating the strong influence of anthropogenic vectors, over-riding natural dispersal rates. One specific example is the arborescent bryozoan *Tricellaria inopinata* (a sessile animal with a short-lived non-feeding larva), which occupied only a short central part of the south coast of England in 1999, but was documented spreading as far as Orkney by 2012 (Dyrynda *et al.*, 2000; Arenas *et al.*, 2006; Cook *et al.*, 2013a; Nall *et al.*, 2014; Bishop *et al.*, 2015b). This illustrates the role that human activities and artificial habitats can have in increasing the spread of some NNS, so that the range of a newly arrived NNS might expand rapidly, despite limited natural dispersal, until reaching conditions currently at the limits of the species' environmental tolerance; thereafter the range limits would be expected to track future environmental changes closely.

Regional differences have also been observed in the distributions of NNS in the UK. Rapid assessment surveys of 61 marinas and harbours around the English coast revealed a greater concentration of sessile animal NNS on the south coast than elsewhere, with low numbers in the north-east (Bishop *et al.*, 2015b) and north-west (Wood *et al.*, 2016). This appears to reflect a wider general gradient in GB, with the prevalence of NNS decreasing south-to-north and a substantial proportion of NNS not yet present in Scotland (Ashton *et al.*, 2006; Nall *et al.*, 2014; Collin *et al.*, 2015). This gradient in part reflects the common arrival of sessile animal NNS on the south coast, in many cases crossing from continental Europe (Bishop *et al.*, 2015a), but it may also indicate a closer matching of current environmental conditions in the south to those in the native ranges of many of the NNS found in this region.

Although valuable insights have arisen from the RAS approach, it should be noted that the funding for this survey work has been largely piecemeal, creating variations in approach, timing and coverage that compromise the ability to draw clear conclusions, particularly in relation to the effects of climate change, from the accumulated data (see Gaps and Challenges section).

3.3 Improvement in understanding the potential future threat from new NNS arrivals

Over the last 5 years, a significant effort has been made in the systematic examination of future potential threats from NNS (i.e., horizon scanning) in Great Britain (Roy *et al.*, 2014a) and Europe (Roy *et al.*, 2015). NNS that were likely to have an impact on native biodiversity within the next 10 years, but were not yet established in the wild were prioritised, using a consensus method involving expert opinion, together with a rapid risk assessment. In the UK study, out of 30 species not native to GB and considered to have a high risk with respect to arriving, establishing and posing a threat to native biodiversity, eight species were classified as 'marine'. Interestingly, the two Asian shore crabs *H. sanguineus* and *H. takanoi* that were ranked in the top ten (see table 2 in Roy *et al.*, 2014a) were found in GB in 2014, shortly after this study was published.

Horizon scanning can provide substantial additional insight into the main threats to GB and adjacent areas in terms of potential new arrivals taking into account predicted environmental change, and the likelihood of NNS already established in southern Europe spreading northwards. This method is heavily reliant though on data availability, particularly on impact, which in many cases is lacking for marine species (Ojaveer *et al.*, 2015) – see Knowledge gaps section - and of course, the accuracy of the predictions resulting from this approach will only be tested over time.

3.4 Development of the Great Britain Non Native Secretariat Information Portal

This Portal provides distribution data for over 3,000 NNS in Great Britain (including terrestrial, freshwater and marine species), such as place or origin and date of introduction and potential methods of introduction. For ~300 species more detailed information is provided, including features for identification, impacts and control methods. The development of this Portal is still ongoing, but is definitely a significant step from a policy and management perspective that has occurred in the last 10 years and an important source of information for research focused on how species distribution patterns can shift as a consequence of climate change.

4 KNOWLEDGE GAPS AND KEY CHALLENGES

4.1 Greater taxonomic expertise on NNS and reliable identifications

For many species recorded in the UK, there has been a dependence on the taxonomic skills of national and international experts. Their ability in understanding current NNS distributions is vital, if future predictions of how climate change may increase their range are to be validated. Significant efforts have been made over the last 10 years in marine NNS identification training by the MBA, SAMS and Invasive Species Ireland, as a decline in the number of these experts was highlighted as a major concern in the past (House of Lords Science and Technology Committee, 2008). Many marine taxa are typically small and difficult to identify (Pederson *et al.*, 2011), and the taxonomy of some marine groups remains relatively poorly resolved (arguably much more so than in terrestrial and fresh-water biota), so fundamental studies are required to determine the true number of NNS and establish valid names for them. Taxa from offshore environments also still receive less attention than near-shore and brackish-water NNS (Ojaveer *et al.*, 2014).

Significant improvements have been made in awareness-raising. Identification literature, online resources, etc. have been produced for a substantial number of NNS in the last decade, although information is still lacking for the majority of NNS recorded in the UK. Efforts by the Non-Native Species Secretariat and partners in the Great Britain NNSs Information Portal have collated and made available information on more than 200 non-native marine and estuarine species either in the UK or with the potential for introduction, including more in-depth fact-sheets and identification cards for many species considered significant for environmental, social or economic reasons. Projects such as the Chinese mitten crab Recording Scheme and Wakame Watch have resulted in resources and web sites aimed at generating records of species, as well as raising awareness of their distribution.

4.2 Requirement for new technologies for the early detection of NNS

The lack of new technologies for the early detection of marine NNS was recently highlighted in a list of the main issues currently facing policy makers (Caffrey *et al.*, 2014; Ojaveer *et al.*, 2014). The detection of species' presence through analysis of 'environmental DNA' (eDNA, such as DNA in water samples or scrapings) or of directly-obtained organismal DNA may significantly enhance surveillance programmes in the future, particularly more enclosed marine systems (i.e., sea lochs, harbours, etc.; Jerde *et al.*, 2011; Dejean *et al.*, 2012). DNA detection has substantial potential in the marine environment (Bourlat *et al.*, 2013). A basic requirement for the technique to be widely used as a method for detecting marine NNS is the development of markers for a large suite of species, together with the resources to process the samples and have the results validated by taxonomic experts.

A key challenge may be obtaining molecular markers with sufficient specificity to enable their widespread (ideally world-wide) use in unequivocally detecting single NNS, particularly given the incompletely-known taxonomy of some speciose groups. The target NNS is likely to be accompanied at a given sampling site by a suite of close relatives that varies in composition geographically. For instance, *Didemnum*, which contains the carpet sea squirt *D. vexillum*, includes almost 240 other species worldwide, several of which may be present in the UK and Ireland. For DNA tests to be developed and used effectively there is a need, not only to ensure rigorous molecular protocols, but also to set taxonomically well-informed quality assurance standards, perhaps under a scheme such as the North East Atlantic Marine Biological

Analytical Quality Control scheme.

4.3 Need for long-term UK-wide monitoring programmes for NNS

Over the last 10 years, baseline data has been recorded from surveys for marine and brackish water NNS throughout the UK (Arenas *et al.*, 2006; Minchin and Nunn, 2013; Nall *et al.*, 2014; Bishop *et al.*, 2015b; Collin *et al.*, 2015; Wood *et al.*, 2015b, 2015c, 2016). This data has been collated by the National Biodiversity Network (NBN), the Archive for Marine Species and Habitats Data (DASSH) and a list of NNS in British marine and brackish waters has been published (Minchin *et al.*, 2013). The GB NNS Information Portal has also gone some way to collating distribution and other information on NNS in GB, including marine species (Roy *et al.*, 2014b). Monitoring programmes, however, for NNS are intermittent and highly dependent on the region (Sambrook *et al.*, 2014). For example, the last region-wide survey in Northern Ireland took place in 2012, whereas for Scotland the surveys have been more ad-hoc and the last detailed country-wide survey took place in 2008 (Cook *et al.*, 2011). In England, the surveys conducted by the MBA have been somewhat more regular (Arenas *et al.*, 2006; Bishop *et al.*, 2015a; Wood *et al.*, 2015b, 2015c, 2016). This is a significant improvement compared with a decade ago, the lack of long-term (i.e., >20 years) data collection, co-ordinated across regions, for the majority of NNS and in-depth biological information, however, continues to mean that the influence of climate change on NNS is difficult to elucidate.

4.4 Need for a greater understanding of the impacts on marine NNS (both established and newly arrived) on native species and habitats and how climate change is contributing to any observed changes.

Evidence is severely lacking to date on the impacts that the majority of marine NNS in the British Isles are having on native species, unlike those seen in the Mediterranean Sea (Galil *et al.*, 2015), and on how climate change is contributing to any observed changes. Some localised impacts have been recorded, including the sporadic poisoning or smothering of farmed organisms in aquaculture and the increased incidence of introduced diseases, such as the herpes virus in *Crassostrea gigas* (Lynch *et al.*, 2012) – see Emerging Issues section for more details and infectious salmon anaemia in Atlantic salmon (Murray *et al.*, 2002), clogging of nets, or fouling of structures – all events of considerable concern for the aquaculture industry in England, Scotland and Ireland, but it is difficult to directly link these impacts to climate change. An example that can be linked to an increase in seawater temperature, however, is the extensive habitat modification to shallow inlets and bays on the south coast of England by *C. gigas* (Herbert *et al.*, 2012).

4.5 Greater modelling capability to predict introductions of NNS either via anthropogenic or natural dispersal for both current and predicted climate change scenarios.

Modelling can greatly assist with describing the most probable routes of natural dispersal from a given starting point and for forecasting the trajectory of spread (route and timing), particularly for species that are highly mobile or have a pelagic larval/ propagule phase that is long-lived. Previous studies have modelled the spread of the invasive lionfish in the Western Atlantic (Johnston and Purkis, 2011) and *Caulerpa taxifolia* in the Mediterranean (Hill *et al.*, 1998) and modelling the risk of the introduction and spread of NNS has been undertaken in the UK and Ireland (Pearce *et al.*, 2012). The models, however, can only give estimates of spread, as they typically use passive particles in their simulations. More recent work has been able to take account of pelagic larval behaviour (Adams *et al.*, 2015) and this approach could greatly enhance the forecasting capability of these models for predicting dispersal patterns of NNS. It is also critical to incorporate human-mediated secondary dispersal into the forecasting models to improve the accuracy of the output.

Predictive models could also be used to ascertain why certain NNS have not been responding to seawater warming in the same way as many others and the role that natural dispersal and anthropogenic vectors play in the secondary spread of a species. For example, the non-native American slipper limpet *Crepidula fornicata* is widespread throughout most European waters following its first introduction in the late 19th century (Crouch, 1894). The species first occurred in Welsh waters in the 1950s in Milford Haven Waterway (Cole and Baird, 1953), where it is now extensively distributed and highly abundant (Bohn *et al.*, 2015). This population, plus a few individuals recorded at Skomer Island, just outside Milford Haven, are the most northerly records along the Welsh coastline, and thus *C. fornicata* has not significantly extended its distribution in over 50 years. The factors limiting further northwards spread of *C. fornicata* from Milford Haven are unclear. The presence of breeding populations of *C. fornicata* in Belfast Lough, Northern Ireland (McNeill *et al.*, 2010) far to the north of the limit in Wales and recent work by Bohn *et al.* (2012) suggests the northward spread is not limited by temperature effects on reproduction or adult mortality, but this work does highlight the critical settlement and early post-settlement process as a potential bottleneck. This, in isolation, or in combination with local and regional hydrodynamics and unsuitable habitat to the north of Milford Haven may be restricting its secondary spread. This highlights the complexities of understanding how NNS react to changes in climate; secondary spread of NNS will be subject to many limiting factors beyond simply those environmental factors associated with global change, which modelling could potentially help to elucidate.

5. EMERGING ISSUES

5.1 Opening of links between the Pacific and Atlantic Oceans due to melting of Arctic sea ice

Climate change will almost certainly alter marine ecosystems globally (Hoegh-Guldberg and Bruno, 2010). Perovich and Richter-Menge (2009) in their review concluded there has been both a significant perennial and end-of-summer decline to the extent of surface sea-ice in the Arctic Ocean. Thinning of the ice cover from ~1.5 – 3m to ~1.0 – 2.25 m has taken place and in some areas there has been a weakening of sea-ice strength (Mooney, 2010). The shrinkage of sea-ice volume and its extent, has resulted in an increase of open water, which enables a greater absorption of the sun's energy. The increase of mean summer air temperatures is greater in the Arctic Ocean than for most world regions (Screen and Simmonds, 2010). Some models predict that there will be an absence of ice by 2037 (Wang and Overland, 2009). Changes such as these are likely to lead to a further increase in sea-temperatures, especially within shallow coastal bays in the Arctic and will also modify oceanographic and biological processes (Greene and Pershing, 2007). Land-sourced runoff and meltwaters are likely to lead to changes in salinity and nutrient runoff and local increases in turbidity (IPCC, 2007).

A significant result of the sea-ice retreat to date has been the appearance of waterways progressively replacing the ice fields. The Northwest Passage, broadly describing the route north of Canada between the North Atlantic Ocean and the Pacific Ocean, and the Northern Sea Route along the coast of Russia have both seen similar ice change. The opening of these waterways means that not only will biological boundaries between the North Atlantic and Pacific Oceans become increasingly blurred as an increase in trans-Arctic migrations is likely (Reid *et al.*, 2009), but the risk of introducing cold-tolerant marine NNS (i.e., species that do not originate from this region) to the Arctic via anthropogenic vectors, such as container vessels, cruise ships and drilling rigs will intensify (Ware *et al.*, 2013).

The last time the Pacific and Atlantic oceans were connected was during the Pliocene which occurred between 1.2 million and 800,000 years ago. During this time, there was a major invasion of new species that had profound impacts on Atlantic ecosystems. For most of the period since the Pliocene the two oceans have been isolated from each other (Reid *et al.*, 2007). In 1998, however, the first recorded movement of a species in modern times between these two oceans was observed, when a substantial retreat of summer ice allowed the Pacific diatom *Neodenticula seminae* to be transferred by currents into the North Atlantic. It bloomed in the Labrador/Irminger Seas, between Canada and Greenland in 1999, and subsequently in 2001 in the Gulf of St Lawrence (Reid *et al.*, 2007). Since then it has spread south to Georges Bank on the eastern coast of the USA and east to south of Iceland (Reid *et al.*, 2009). Its recent re-appearance in the North Atlantic could, therefore, signify the start of a new influx of organisms to the North Atlantic.

A further consequence of the retreating ice fields in the Arctic is the increasing extent of navigable waterways, which are already having a significant impact on global shipping. For example, since the mid-1970s, the number of transits through the Northwest Passage has quadrupled, reaching 20 per year in 2009 – 2011, including relatively slow moving ice-breakers, passenger ships, tugs and supply vessels (GNWT, 2013). Longer distance transportation of supplies, however, is predicted as these new waterways could potentially open up sea routes between Europe and Asia that are thousands of miles shorter than existing ones and could reduce the travel distance between Rotterdam and Yokohama by 40%, and from Rotterdam to Seattle by 25% (NATO, 2010). To demonstrate the feasibility of using the Arctic for global shipping, in September 2009 two German commercial ships, accompanied by two Russian icebreakers, successfully sailed the Northern Sea Route through the Northern Passage and delivered goods to Japan, South Korea and Russia. This route reportedly shortened their journey by 4,000 km (GNWT, 2013). The opening of these new shipping routes would potentially create new pathways for species known to be invasive elsewhere in the world (Goldsmit *et al.*, 2014). For example, the Northern Pacific starfish *Asterias amurensis* is highly invasive in Australia and Tasmania where it was introduced through the transfer of larvae in ship ballast (Byrne *et al.*, 1997). It is native to and common in the Northern Pacific, as far North as the Barents Sea. A rise in shipping between this region and the North of Europe would increase the potential for the introduction of this species to British waters.

A major vector for the transmission of NNS is via commercial vessel hulls and ship sea-chests (used during ballast water exchange), either as sessile (fouling), boring, vagile, or clinging species (Minchin, 2006). Alien species introduced by these vectors involve a wide range of taxa, including: macroalgae (e.g., *Undaria pinnatifida*, *Asparagopsis armata*), barnacles (e.g., *Austrominius modestus* and *Amphibalanus amphitrite*), bryozoans (e.g. *Tricellaria inopinata* and *Watersipora subatra*), and ascidians (e.g., *Didemnum vexillum*, *Botrylloides violaceus*, and *Styela clava*). Most of these species are commonly found in ports and marinas throughout Europe (Arenas *et al.*, 2006; Ashton *et al.*, 2006; Griffith *et al.*, 2009; Cook *et al.*, 2013b; Bishop *et al.*, 2015b) and slow moving vessels (e.g., barges, drilling rigs and ice-breakers) or ships berthed in one place over long periods are particularly prone to transmitting alien species. Ballast tanks (including ballast water, ballast sediment and species attached to the tank walls) have also been highlighted in numerous studies as an important transmission vector for alien species, particularly phytoplankton (Pederson et al., 2011; Minchin et al., 2013). It has been suggested that a number of *Bacillariophyta* (diatoms), such as *Odontella sinensis* and *Coscinodiscus wailesii*, had been introduced via ballast water (Eno *et al.*, 1997). The predicted increase in global shipping through Arctic waters, therefore, is highly likely to exacerbate the problem of an exchange of alien species between the Atlantic and Pacific Oceans and the ecological impacts that will ensue.

To conclude, it is highly likely that substantial increases in commercial shipping in northern high latitudes will occur over the next decade, as a consequence of sea ice retreat and the resulting access to shorter sea routes between the Pacific and Atlantic oceans (compared with travelling via the Suez Canal). The physical barrier of the Arctic ice shelves will continue to be breached at an ever increasing pace and species introduction and establishment either via natural dispersal or anthropogenic vectors will mean that this already vulnerable region will become ever more prone to the ecological and economic impacts caused by marine NNS.

5.2 Climate change, marine invasions and diseases

Marine macro-organisms are usually associated with a varied microbial community, forming with their host the "holobiont" community. Members of these symbiotic communities (i.e., eukaryotes, bacteria and viruses) can be beneficial, neutral or detrimental to their hosts. The diversity of these "microbes" is only beginning to be understood thanks to the development and wider application of new generation sequencing methods. Transport, introduction and potential spread of these symbiotic microbial species along with their hosts have so far usually remained unnoticed.

An exception is the case of parasites and pathogens transported with aquaculture stock. It usually involves the spread from a symptom free carrier to new hosts (from different strains or species). There are many examples in the oyster industry, with for instance the parasitic copepods *Myicola ostreae* and *Myticola orientalis*, the protozoa *Bonamia ostreae* and *Marteilia refringens* and an irido-like virus, all brought with oyster stock from the Northern Pacific into Europe (see supporting information in (Mineur *et al.*, 2014) for complete case reviews). The most recent case of such imported disease is the ostreid herpesvirus (Mineur *et al.*, 2015b). Every summer since 2008, a particular variant (called the OsHV-1 μVar) has caused a lot of mortalities in European stock of *Crassostrea gigas*. Studies have shown that temperatures above 16 °C constitute an important factor triggering pathogenicity (mortality of the host) and virulence (concentration of the virus) (Pernet *et al.*, 2015). In the UK, mortality outbreaks due to this virus have been recorded only on the south coast of England, in Devon, Kent and Essex (Renault, 2011; AHAW, 2015; CEFAS, 2015a;). Rising temperatures, however, are likely to cause a northward increase in such mortalities in the near future and global warming is more likely to exacerbate and increase the frequency of such episodes.

Little is known about the origin and possible introduction of marine pathogens in the marine environment. It is probable however, that a large and diverse amount of 'microbes' are associated with their host organisms. Even if the NNS do not become established, parts of their holobiontic communities may be able to survive and thrive by quickly switching to native hosts. Therefore, native communities, already exposed to global warming, as well as different local stressors, may experience an even stronger selection pressure with these new arrivals. This would be especially true in regions that include hubs of introductions, such as ports and aquaculture installations.

5.3 Marine Debris/ Litter - a newly recognised vector for warm-water species?

Rafting of marine species by attachment to natural objects, such as tree debris and macroalgae, is a natural method of species dispersal. For suitably equipped species, dispersal potential due to natural rafting is often limited by the ability of the raft object to persist in the marine environment without eroding or sinking (Thiel and Gutow, 2005a). Plastic items have a far greater potential to persist in the marine environment than natural objects, increasing the distance they could potentially travel. Holmes *et al.* (2015) report that in 2013 – 2014, prolonged westerly gales resulted in large numbers of rafting species being stranded on the south-west coasts of the British Isles. Whilst the authors primarily reported on six species of gastropod, they also mention species from a number of other taxa. At least one of these, the bivalve mollusc *Pinctada imbricata imbricata* has been recorded previously on stranded flotsam in UK waters (Turk, 1988) and is very closely related to the Indo Pacific sub-species *P. i. radiata*, which is invasive in the Mediterranean Sea (Lodola *et al.*, 2013). Whilst there is no evidence of species becoming established in the British Isles having arrived by rafting, it is possible that given the appropriate conditions some of the species introduced could become established. Thiel and Gutow (2005b) suggest that rafting might only be a significant evolutionary process if rafting species are capable of establishing new populations on reaching new habitats. Additionally, they identify the ability of rafting species to survive extreme conditions. Such attributes would increase the potential for establishment. Many of the species identified are warmer water species and could potentially benefit from warming seas.

5.4 Belief-driven animal release

Climate change increases the risk that live animals originating from a warmer region of the globe which are deliberately released from captivity may survive and establish populations. The practice of 'life release' or 'mercy release' of captive live animals, including marine forms such as crabs, lobsters and bivalves, is widespread in Buddhist communities worldwide and is ostensibly to save the animals' lives and thereby obtain good karma for the releaser (Shiu and Stokes, 2008). This can have the unintended consequence of introducing NNS to the wild (Liu *et al.*, 2012). The animals to be released are generally supplied expressly for this purpose, and the substantial commercial demand this creates can lead to the culture of exotic species or their capture and importation from abroad simply to supply the demand, followed by release outside of their native range (Shiu and Stokes, 2008). A release of 200 crabs (likely to be *Cancer magister*) and the lobster *Homarus americanus* into the sea close to Brighton, southern England, in 2015 highlighted the potential for this vector to result in the introduction of numerous species (Walker, 2015). Similarly, climate change would be expected to increase the risk associated with the release of exotic commercial species by animal rights activists.

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